

Modeling the Transport and Inactivation of *E. coli* and Enterococci in the Near-Shore Region of Lake Michigan

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To investigate the transport and fate of fecal pollution at Great Lakes beaches and the health risks associated with swimming, the near-shore waters of Lake Michigan and two tributaries discharging into it were examined for bacterial indicators of human fecal pollution. The enterococcus human fecal pollution marker, which targets a putative virulence factor—the enterococcal surface protein (esp) in *Enterococcus faecium*, was detected in 2/28 samples (7%) in the tributaries draining into Lake Michigan and in 6/30 samples (20%) in Lake Michigan beaches. This was indicative of human fecal pollution being transported in the tributaries and occurrence at Lake Michigan beaches. To understand the relative importance of different processes influencing pollution transport and inactivation, a finite-element model of surf-zone hydrodynamics (coupled with models for temperature, *E. coli* and enterococci) was used. Enterococci appear to survive longer than *E. coli*, which was described using an overall first-order inactivation coefficient in the range 0.5–2.0 per day. Our analysis suggests that the majority of fecal indicator bacteria variation can be explained based on loadings from the tributaries. Sunlight is a major contributor to inactivation in the surf-zone and the formulation based on sunlight, temperature and sedimentation is preferred over the first-order inactivation formulation.

Introduction

Millions of people visit the 500 plus recreational beaches in the Great Lakes every year. Recently, water quality degradation due to fecal bacteria at the shoreline has been increasing,

and thus, the potential risk to human health is of interest to beach managers and the public. There were more than 1800 (combined) days of closings and advisories caused by pollution at Great Lakes beaches in 2003 (32% more than in the previous year), and in 2004 the number went up to 3081 (1). The closings and advisories at Michigan Great Lakes beaches were due to elevated fecal bacteria levels from unknown sources of contamination (1). Fecal pollution of such recreational waters has been associated with gastrointestinal diseases and infections (e.g., eye or respiratory) in beachgoers (2). Water quality at these beaches is impacted by a variety of fecal pollution sources (e.g., agriculture, human, wildlife), and the indicators, such as *Escherichia coli* (EC) and enterococci (ENT), that are used to evaluate recreational water quality are found in a variety of mammalian hosts. Thus, they give no indication as to the source of fecal pollution. Identification of fecal pollution sources is important in assessing the risk to human health associated with water use and for the management and remediation of beaches and contamination sources. Microbial source-tracking, coupled with process-based modeling, has the potential to test hypotheses about sources of contaminants. One tool for identifying human-sourced fecal pollution is the presence of a putative virulence factor, the enterococcal surface protein (esp) in *Enterococcus faecium*. This marker has been found only in *Enterococcus faecium* isolated from fecal samples of human origin, and the presence of this marker is thus indicative of human fecal pollution (3).

Understanding factors that influence transport of fecal pollution along the shoreline will enable us to quantify risks to human health from recreational use of these beaches. Besides human health risks, beach closures have significant negative impact on local economies. Traditional techniques used for recreational beach monitoring require an incubation period of 24 h for the assays; however, EC concentrations are known to vary significantly even in a short period. Therefore, predictive modeling has been suggested as an alternative to the current practice based on measurements. Past efforts to develop mechanistic models for pathogenic organisms in surface waters focused mainly on marine coastal waters (4–6), and relatively few studies examined large freshwater systems such as the Great Lakes. Due to the complexity of the processes and the large number of variables involved, statistical approaches have been used to predict beach closures (7). Modeling near-shore, wind-driven circulation and the transport of chemical and biological agents is challenging due to interactions with complex lake-wide circulation (8). The goals of this study were to (a) identify whether human fecal pollution may be impacting the Great Lakes beaches, (b) develop a near-shore transport model for fecal pollution in Lake Michigan and to identify key processes influencing inactivation, and (c) see if the observed variability in the concentrations can be explained based on loadings from Trail Creek and Kintzele Ditch.

Description of Sites

The region of our primary interest included approximately 72 km of shoreline within the state of Indiana (Figure 1). The watershed that contributes to the shoreline encompasses the Porter, Lake, and LaPorte counties. The cities of Gary, Hammond, East Chicago, and Michigan City along the shoreline are the major population centers in the watershed. The monitoring data were focused on the locations close to Michigan City. There are three main tributaries and a small ditch flowing into Lake Michigan in the domain of interest: Indiana Harbor Canal at East Chicago (USGS 04092750),

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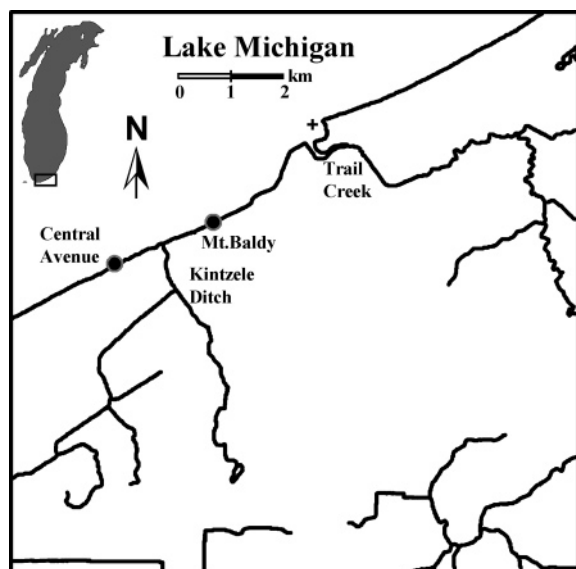


FIGURE 1. Map of southern Lake Michigan showing the Indiana shoreline.

Portage-Burns Waterway (Burns Ditch, USGS 04095090), Trail Creek at Michigan City Harbor (USGS 04095380), and Kintzele Ditch. Other creeks flowing into Lake Michigan contribute little discharge and were not considered in our modeling. For transport simulations, it was important to quantify the strengths of various loadings, which are “beachshed dependent”. Potential sources in the near-shore environment include point sources (e.g., wastewater treatment plants), combined sewer overflows (CSOs), nonpoint sources (e.g., faulty septic systems, manure storage, wildlife), and tributaries themselves. Most of the CSOs discharge into the rivers which eventually drain into the lake but few CSOs discharge directly to the shoreline; therefore, CSOs were not considered as sources in this study. Tributaries that enter Lake Michigan within Indiana are considered the most significant source of EC and ENT to the shoreline. Septic systems were not considered in our modeling due to lack of information on the areas with high septic vulnerability along the shoreline. Previous research shows that contamination at beach sites in Lake Michigan is mainly associated with localized sources of pollution. Based on historical water-quality monitoring data, Kintzele Ditch and Trail Creek were considered as the pathogen sources out of the four main tributaries. For the hydrodynamic model, however, all four tributaries discharging into Lake Michigan were considered.

Materials and Methods

Sampling took place in the summer of 2004 (July–August). Samples were collected and analyzed for fecal pollution indicator bacteria (EC and ENT) and for the presence of the putative human fecal pollution source marker (the marker that targets the *esp* gene in *Enterococcus faecium*) (3). Water samples were collected from knee-deep water (~45-cm depth) from two sites at Central Avenue Beach, three sites at Mt. Baldy Beach, and one site each at the mouths of Kintzele Ditch and Trail Creek. Solar insolation was measured using a Campbell pyranometer near Trail Creek (+ Figure 1). EC were evaluated in water samples using membrane filtration according to section 9222G (9). Membrane filters were incubated on mFC agar at 44.5 °C for 24 h, transferred to EC-MUG agar (Difco, 222200), and incubated for 24 h at 44.5 °C. Individual colonies that produced fluorescence under a long-wavelength (366-nm) ultraviolet light were considered EC. Colony-forming units (CFU) per 100 mL were recorded.

ENT were isolated on mEI agar according to U.S. EPA Method 1600, incubated at 41 °C, and CFU enumerated after 24 h. Membrane filters that contained approximately 50 or more CFU were assayed for the presence of the *esp* gene in *Enterococcus faecium* using primers specific to bacteria of human origin (3). Bacterial colonies were removed from the membrane filters by suspending the filters in 10 mL of tryptic soy broth and incubating for 2 h at 41 °C. DNA was extracted from 1 mL of this suspension using the Qiagen QIAamp DNA Mini Kit according to the manufacturer’s directions (cell lysis through DNA purification). The forward primer, which is specific for the *E. faecium esp* gene used, was (5′-TAT GAA AGCAAC AGCACA AGTT-3′) (3). A conserved reverse primer (5′-ACG TCG AAA GTT CGA TTT CC-3′) was used for all reactions (10). PCR reactions contained 1X PCR buffer, 1.5 mM MgCl₂, 200 μM of each dNTP, 0.3 μM of each primer, 0.5 U of HotStarTaq DNA polymerase (Qiagen), and 1 μL template DNA per 20 μL of reaction. Amplification was performed with an initial step at 95 °C for 15 min followed by 35 cycles at 94 °C for 1 min, 58 °C for 1 min, 72 °C for 1 min, and a final extension at 72 °C for 7 min. PCR products were separated on a 1.5% agarose gel stained with GelStar nucleic acid stain (BioWhittaker) and viewed under UV light.

Modeling

Interactions between the near-shore and lake-wide circulation are important; hence, our computational domain included the entire Lake Michigan (Figure 2). To resolve the shoreline accurately, a finite-element model was used. The finite-element mesh was gradually refined from a resolution of approximately 1–2 km for the whole lake to about 100 m near-shore. Major considerations in the model development included the representation of shoreline and bathymetry, meteorological data including wind direction and speed, solar insolation, hydrological flows, water temperature, and loadings of EC and ENT. Bathymetry data with a resolution of 3 arc-seconds were obtained from the NOAA National Geophysical Data Center. Figure 2 shows the details of the finite-element mesh and the near-shore bathymetry. Hourly meteorological data (air and dew point temperatures, barometric pressure, wind speed and direction) were obtained from the NOAA National Climatic Data Center. Water-level data were obtained from the Center for Operational Oceanographic Products and Services (CO-OPS), NOAA, and the National Data Buoy Center. The finite-element mesh consisted of 8-noded-quadrilateral and 6-noded-triangular elements with a total number of 11,716 elements (30,829 nodes). Wind fields and initial conditions for temperature were obtained by interpolating data from several stations in Michigan (Harbor Beach, Rock Cut, Ludington, Muskegon), Indiana (Michigan City), and Illinois (Calumet Harbor, Chicago) as well as from two NDBC buoys in Lake Michigan (45002 and 45007). Wind data measured at different heights were reduced to a common 10 m height using a method proposed by Smith (11).

Since our primary interest was in the near-shore region where depths are shallow, we employed a vertically integrated hydrodynamic model based on the finite-element model RMA10 (12) to describe the wind-driven circulation in Lake Michigan. Water quality models for temperature, EC, and ENT were coupled to the hydrodynamic model. A 1-month period (July 12–August 13, 2004, Julian days 194–226) that corresponded to our sampling period was selected for the modeling. The vertically integrated hydrodynamic equations in *x* and *y* directions are shown below (12)

$$\frac{\partial h}{\partial t} + U \frac{\partial h}{\partial x} + V \frac{\partial h}{\partial y} + h \left(\frac{\partial U}{\partial x} + \frac{\partial V}{\partial y} \right) = 0 \quad (1)$$

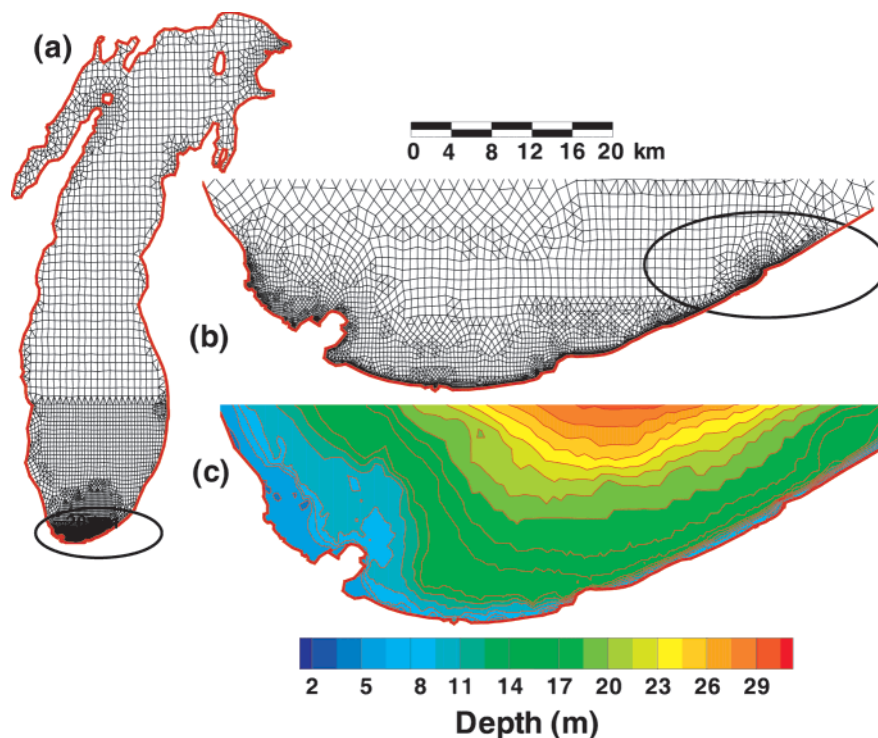


FIGURE 2. (a) Finite-element mesh of Lake Michigan showing (b) the Indiana Shoreline and (c) the near-shore bathymetry.

$$h \frac{\partial U}{\partial t} + hU \frac{\partial U}{\partial x} + hV \frac{\partial U}{\partial y} - fVh = \frac{1}{\rho} \left[\frac{\partial}{\partial x} \left(\bar{\epsilon}_{xx} h \frac{\partial U}{\partial x} \right) + \frac{\partial}{\partial y} \left(\bar{\epsilon}_{xy} h \frac{\partial U}{\partial y} \right) \right] - gh \left(\frac{\partial a}{\partial x} + \frac{\partial h}{\partial x} \right) - \frac{Ugn^2 \sqrt{(U^2 + V^2)}}{h^{1/3}} + \zeta W^2 \cos \psi \quad (2)$$

$$h \frac{\partial V}{\partial t} + hU \frac{\partial V}{\partial x} + hV \frac{\partial V}{\partial y} + fUh = \frac{1}{\rho} \left[\frac{\partial}{\partial x} \left(\bar{\epsilon}_{yx} h \frac{\partial V}{\partial x} \right) + \frac{\partial}{\partial y} \left(\bar{\epsilon}_{yy} h \frac{\partial V}{\partial y} \right) \right] - gh \left(\frac{\partial a}{\partial y} + \frac{\partial h}{\partial y} \right) - \frac{Vgn^2 \sqrt{(U^2 + V^2)}}{h^{1/3}} + \zeta W^2 \sin \psi \quad (3)$$

where U, V are the depth-averaged velocities in the x, y directions, h is the water depth, a denotes the bottom surface elevation, g is the acceleration due to gravity, W is the wind velocity, ψ is the wind direction, ζ is an empirical wind coefficient, and f is the Coriolis parameter. n denotes the Manning's roughness coefficient, and $\bar{\epsilon}$ is the depth-averaged eddy viscosity. A constant value of $2.0 \text{ m}^2/\text{s}$ was used for the eddy viscosity in the near-shore region. Viscosity values in the offshore region were variable and depended on the element size and the velocity gradients as described by the Smagorinsky formulation. Equations for the transport of EC, ENT, and temperature have the following general form

$$\frac{\partial(hC)}{\partial t} + U \frac{\partial(hC)}{\partial x} + V \frac{\partial(hC)}{\partial y} = \frac{\partial}{\partial x} \left(D_{xx} h \frac{\partial C}{\partial x} + D_{xy} h \frac{\partial C}{\partial y} \right) + \frac{\partial}{\partial y} \left(D_{yx} h \frac{\partial C}{\partial x} + D_{yy} h \frac{\partial C}{\partial y} \right) - khC \pm G \quad (4)$$

where C denotes the depth-averaged temperature or the concentrations of EC or ENT (CFU/100 mL), and D_{xx} , D_{xy} , D_{yy} , and D_{yx} are the depth-averaged dispersion and turbulent diffusion coefficients. In (4), G is a general term that denotes sources and sinks. Equation 4 is coupled to the hydrodynamic model and was solved using RMA11 (12). For the temperature equation, G included contributions due to the net shortwave radiation, the longwave back radiation, evaporation, con-

densation, the heat flux due to sensible heat transfer, and heat inputs from tributaries. Details of these different fluxes and the ranges of parameters have been described by Martin and McCutcheon (13). Initial conditions for the model assumed that the lake was at rest at time $t = 0$. A background value of 3 CFU/100 mL was used as the initial condition for both EC and ENT based on our observations. Observed water temperatures at different stations were interpolated to the finite-element mesh to create an initial condition for the thermal model. Boundary conditions for the hydrodynamic model included the no leakage condition across the surface and the bottom, zero pressure and wind stress at the free surface, and drag at the bottom surface.

Numerous factors influence the fate, transport, and persistence of EC and ENT. These include sunlight, nutrient content, suspended solids concentration, removal by sedimentation, water temperature, pH, and predation. EC and ENT could potentially replenish at night time (14) either due to the recovery of nonculturable cells or due to unidentified sources. The presence of suspended solids in the water column has been shown to increase EC survival rates by limiting the effects of sunlight (15). EC survive longer in turbid conditions (16), and both EC and ENT survive longer in cold temperatures than in warm temperatures (17). Increased death rates at the higher temperatures may be due to damage to the bacterial cell components or due to increased predation (18). For the enteric bacteria, the effects of temperature have been reported to be less important in the presence of light (16). Many studies have also reported that the inactivation of EC is more rapid in saline waters than in freshwater (19). An important question often addressed in the literature is related to the differences in disappearance rates for different bacterial strains. For example, Menon et al. (20) did not find a significant difference in the rates of disappearance of the strains tested (*E. coli*, *S. faecium*, and *S. typhimurium*). Sinton et al. (18), on the other hand, reported a disappearance rate for EC that is four times higher compared to ENT. After examining the important inactivation mechanisms reported in the literature and their mathematical formulations, we used two different formulations for inactivation: (a) an overall

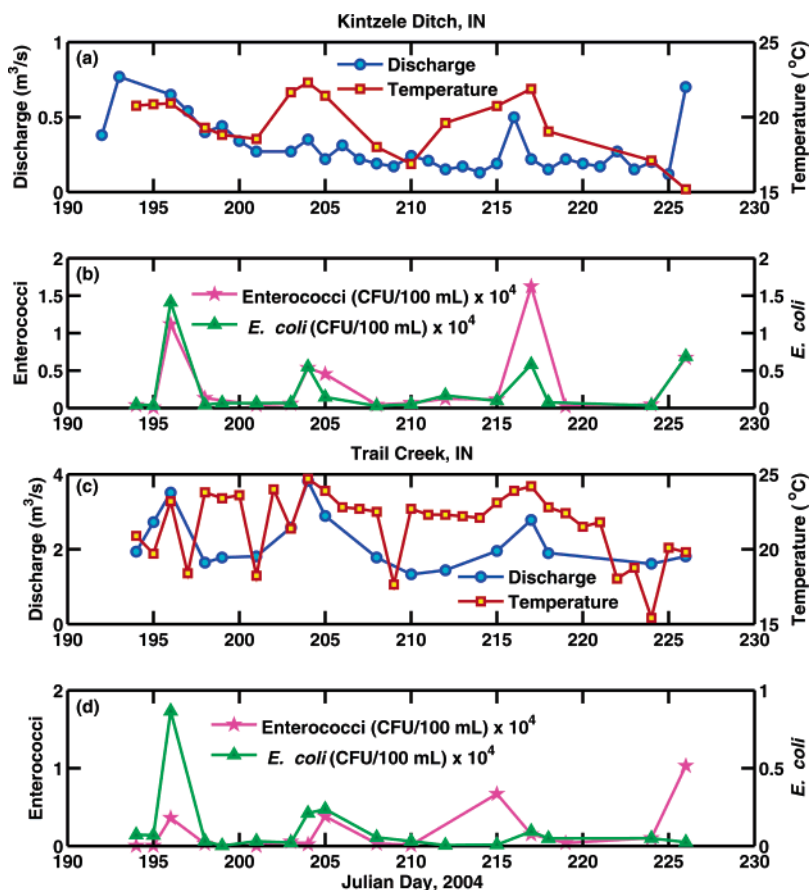


FIGURE 3. Discharges and inflowing concentrations from Kintzele Ditch and Trail Creek during the summer of 2004.

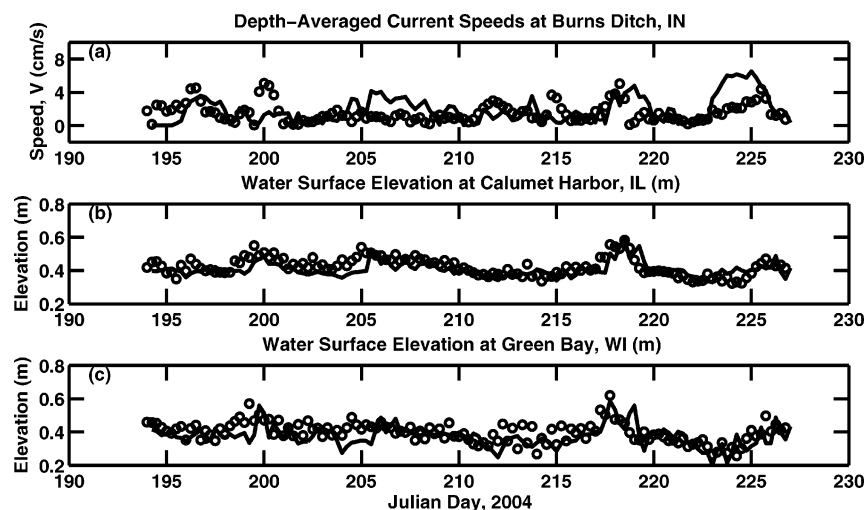


FIGURE 4. Comparison between observed (symbols) and simulated (lines) hydrodynamic variables. (a) Depth-averaged currents at Burns Ditch, IN. Symbols show the ADCP data. (b) Water surface elevation at Calumet Harbor, IL and (c) water surface elevation at Green Bay, WI.

first-order inactivation rate that did not depend on temperature, light, or settling and (b) a time-dependent inactivation rate based on temperature, sedimentation, and observed solar insolation as shown below (5)

$$k(I, T, v_s) = \left(f_p \frac{v_s}{H} + k_l I(t) \right) \theta^{(T-20)} \quad (5)$$

where $k(I, T, v_s)$ is the overall inactivation rate, k_l is the inactivation rate for light ($\text{W}^{-1} \text{m}^2 \text{d}^{-1}$), $I(t)$ is the measured solar insolation (Wm^{-2}) as a function of time, θ is a temperature correction factor (usually 1.07 (21)), f_p is the

fraction of pathogens attached to the suspended sediment, v_s is the settling velocity, and H is the water column depth. The above formulations provided quantitative estimates of the overall inactivation in addition to allowing us to explore further the relative importance of the effects of light, temperature, and sedimentation during summer conditions in Lake Michigan.

Results and Discussion

Discharge, temperature, and concentrations of EC and ENT from tributaries flowing into the study region are shown in Figure 3. The hydrodynamic model was calibrated using data

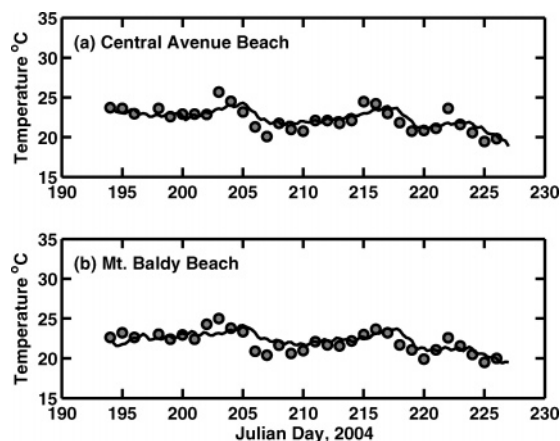


FIGURE 5. Observed and simulated water temperatures at two beaches in the near-shore region of Lake Michigan.

obtained from a 1200 kHz Acoustic Doppler Current Profiler (ADCP) deployed at Burns Ditch during summer 2004 as well as water-surface elevations for several stations throughout the lake. These comparisons are shown in Figure 4. Results from the hydrodynamic model were generally found to be consistent with known circulation patterns in southern Lake Michigan (8). While the water surface elevations are simulated fairly accurately, simulating near-shore currents was relatively challenging due to the lack of high resolution coastline and bathymetric data required to describe currents accurately. Our simulated currents were of the same order of magnitude as the observed currents, and some errors are likely involved in averaging the ADCP data as well. Overall, the hydrodynamic model provided a reasonable description of currents and water surface elevations. Comparisons between the observed and simulated temperatures (Figure 5) showed that the vertically integrated model was able to describe the transport mechanisms of advection and dispersion reasonably well.

Comparison of observed and simulated EC and ENT counts at Mt. Baldy Beach are shown in Figures 6 and 7 for different first-order inactivation rates as well as the formulation shown in eq 5. Mt. Baldy is approximately mid-way between Trail Creek and Kintzele Ditch. Pollution is generally carried along-shore in an eastward direction, but simulations

indicated that currents often reversed their direction, resulting in pollutant plumes that traveled in the opposite direction as well. As a result, observed EC and ENT counts at Mt. Baldy were influenced by loadings originating from both Trail Creek and Kintzele Ditch. Results for $k=0$ (no inactivation) indicate that the observed data cannot be described based on advection and dispersion alone. Both EC and ENT can be described using an overall first-order rate. k values between 0.5 and 0.8 d^{-1} described the observed EC concentrations at Mt. Baldy. However, values around 0.5 or less better described the inactivation of ENT. This can be seen clearly from the cumulative probability plots in Figure 7. Published k values for EC in freshwater typically range from 0.72 to 1.44 d^{-1} (20, 21). Our results suggest that, in Lake Michigan, the lower range of the decay rates should be employed for describing the peak values which are important from the point of beach closures. Figure 7 shows that two different first-order inactivation rates were required to describe the higher and lower (or background) concentrations. For example, it appears that $k = 1.5 \text{ d}^{-1}$ was required to describe ENT values less than 50 CFU/100 mL, while values less than 0.5 d^{-1} described the higher concentrations better. This is attributed to the inability of the first-order formulation to describe the entire range of variability in inactivation. Our results indicate that ENT survive longer in Lake Michigan compared to *E. coli*. This implies that if enterococci was used as an indicator organism as opposed to *E. coli*, then more unsafe-water notifications would result.

Comparisons based on the light-dependent rates showed a similar pattern as obtained from an overall first-order rate (Figures 6 and 7). For EC, the light-dependent rate produced a slightly better description (RMSE = 0.808) compared to the first-order rate with $k = 0.5$ (RMSE = 0.835), but $k = 0.8$ and $k = 2.0$ produced better overall agreement with data (RMSE: 0.770 and 0.705, respectively). The RMSE values are somewhat biased by the relatively large number of low concentrations in the observed data. The light-dependent inactivation results shown in Figure 7 used a k_l of $0.0026 \text{ W}^{-1} \text{ m}^2 \text{ d}^{-1}$, a fraction f_p of 0.1 (4), and a v_s value of 5 m d^{-1} (22). These numbers represent one of several possible scenarios that describe the peak concentrations reasonably well. Although we did not attempt to find the best parameters in the light-based formulation that minimize the RMSE, this formulation is more general. Comparison of the relative magnitudes of various

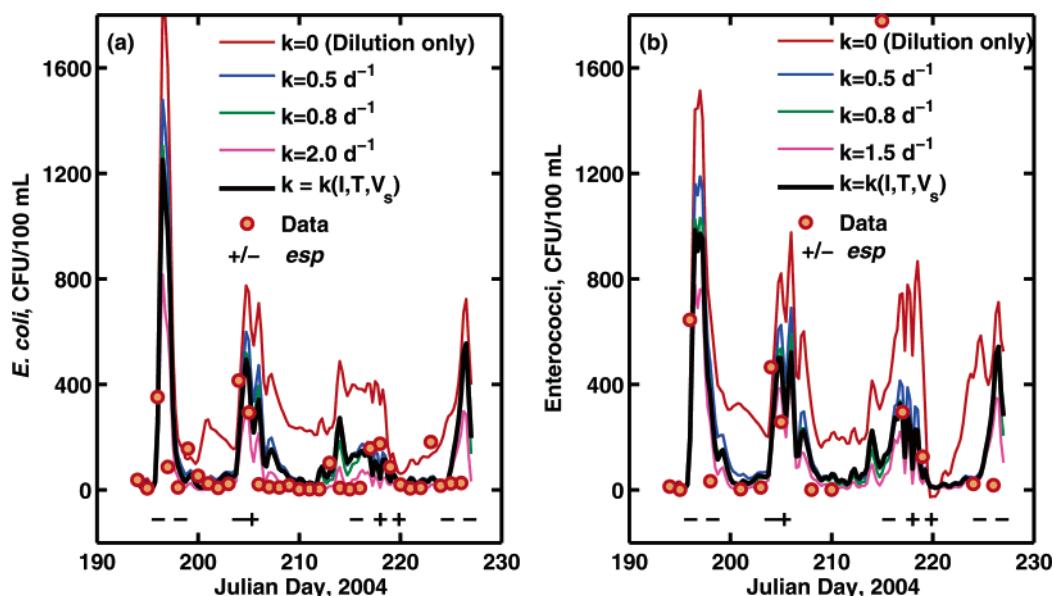


FIGURE 6. Comparison of observed and simulated counts of (a) *E. coli* and (b) enterococci for a 1-month period during summer 2004 at the Mt. Baldy Beach.

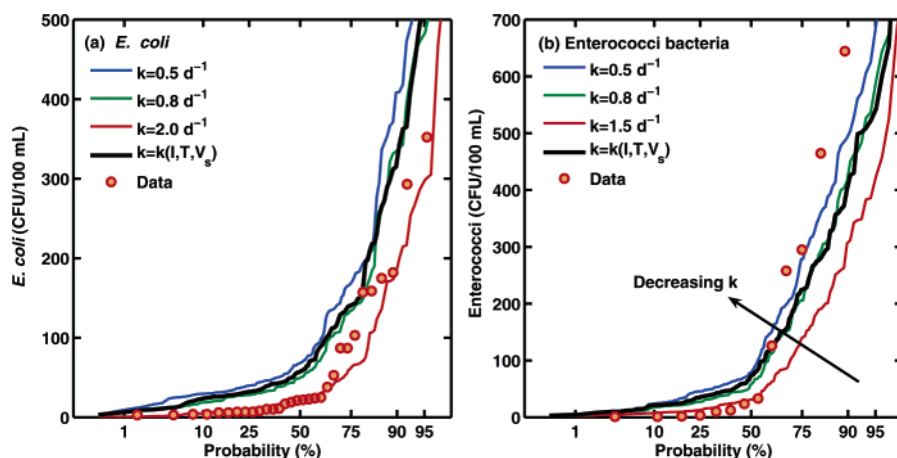


FIGURE 7. Cumulative probability plots showing observed and simulated concentrations for (a) *E. coli* and (b) enterococci.

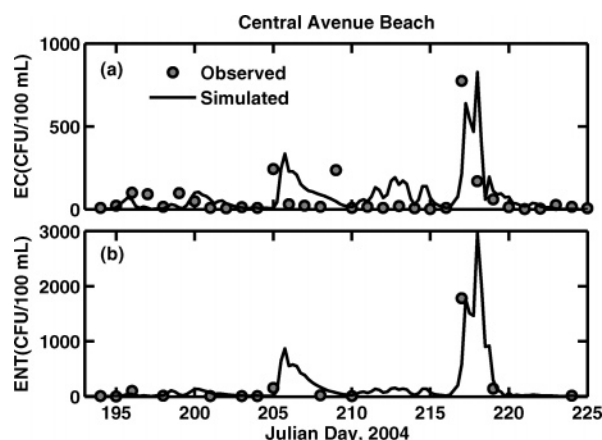


FIGURE 8. Comparison of observed and simulated counts of (a) *E. coli* and (b) enterococci at the Central Avenue Beach.

terms in eq 5 showed that inactivation was primarily light-controlled, and temperature and settling effects were relatively weak for this 1-month period. Figure 8 shows the comparisons for EC and ENT at the Central Avenue Beach based on the light-based inactivation formulation and the same set of parameters used earlier for the Mt. Baldy Beach. The model was able to describe the observed data well (RMSE: 0.83 for EC and 0.55 for ENT) which indicates that the parameters are probably reasonable.

Fecal contamination from sewage is a significant public health concern due to the known presence of human viruses and parasites in these discharges. The *esp* human pollution marker was found (Figure 6) on three occasions (Julian days 205, 217, 218), suggesting that a human fecal pollution source was partially contributing to the fecal contamination at Mt. Baldy Beach on these dates. On Julian day 218, there was a large variation between the observed and simulated ENT peak concentrations at Mt. Baldy (Figure 6b). This suggests that, at one location on the beach, there was a significant input of fecal contamination that was not likely to be sourced from Trail Creek or Kintzele Ditch. Detailed information related to EC sources and loadings is required to improve model predictions, particularly the variability in the background concentrations. The Indiana recreational water quality standards require that the geometric mean of 5 samples over a 30-day period is less than 125 CFU/100 mL, with no sample testing higher than 235 CFU/100 mL. The model was able to predict (and the observed data showed) that Mt. Baldy Beach was above the Indiana standard for full-body contact for recreational waters approximately 10% of the time (Fig-

ure 7a). Recently, a rapid method for detection of ENT has been used (2). Our results suggest that ENT is a viable indicator for freshwater beaches in Lake Michigan, based on transport and fate. Measurements of in situ die-off rates will provide additional support to this observation. The use of the sewage marker demonstrates the risk associated with human sewage, and the enhanced persistence of ENT compared to EC suggests this to be a superior indicator of pollution. While other factors need to be investigated and incorporated into the model to improve these predictions, the findings shown here indicate the usefulness of modeling the near-shore environment using this approach and in being able to predict, and thus potentially reduce, threats to human health from use of these recreational waters.

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Supporting Information Available

Additional data and analysis including text, tables, and figures. This material is available free of charge via the Internet at <http://pubs.acs.org>.

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